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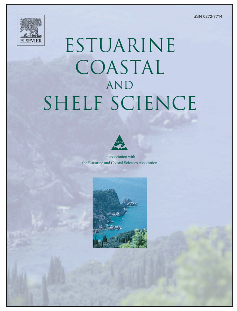


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# Geospatial modelling and map analysis allowed measuring regression of the upper limit of *Posidonia oceanica* seagrass meadows under human pressure

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**Abstract**

Marine coastal ecosystems are facing structural and functional changes due to the increasing human footprint worldwide, and the assessment of their long-term changes becomes particularly challenging. Measures of change can be done by comparing the observed ecosystem status to a purposely defined reference condition. In this paper, a geospatial modelling approach based on 2D mapping and morphodynamic data was used to predict the natural position of the upper limit (i.e., the landward continuous front) of *Posidonia oceanica* seagrass meadows settled on soft bottom. This predictive model, formerly developed at the regional spatial scale, was here applied for the first time at the Mediterranean spatial scale in eight coastal areas of Spain, France, Italy, and Greece showing different coastal morphologies and hydrodynamic characteristics, and affected by a number of natural and/or human local disturbances. The model was effective in measuring the regression (i.e., seaward withdrawal) of the meadow upper limit. In all the meadows investigated the upper limit was regressed, laying deeper than the reference condition, with the proportion of regression ranging from 17.7% to 98.9%. The highest values of regression were found in Spain and in France, and were consistent with the highest levels of fragmentation detected with map analysis and of coastal pressures. This geospatial modelling approach represents an effective tool to define the reference conditions when proper pristine areas or historical data are not available, thus allowing the assessment of long-time changes experienced by seagrass ecosystems due to human impacts.

**Keywords:** seagrass, predictive modelling, reference conditions, morphodynamics, *Posidonia oceanica*, Mediterranean Sea.

## Introduction

During the last century, the incessant urban development of coastal zones caused radical changes in marine ecosystems and a constant decline of their biodiversity (Benoit and Comeau, 2005; Shochat et al., 2006). Quantification of these long-term changes represents one of the main goals of on-going research on management and conservation of coastal marine ecosystems (Halpern et al., 2015; de Andrés and Barragán, 2016).

In European waters, the Marine Strategy Framework Directive (European Council, 2008) imposes to all EU Member States the maintenance (or restoration) of the "good environmental status" and the seafloor integrity of their water bodies by 2020. These objectives can be achieved once the status of ecosystems has been evaluated through the adoption of specific bioindicators, descriptors and related ecological indices (Borja et al., 2010, 2013; Personnic et al., 2014; Rastorgueff et al., 2015; Piazzi et al., 2017). Then, evaluation of changes over time requires the comparison of the current status with previous baselines (i.e., the reference conditions), which may represent the ecosystem status before heavy human impact (Duarte et al., 2008; Borja et al., 2012). Reference conditions can be retrieved using: i) historical information; ii) pristine sites (i.e., natural areas with little or no human pressures) or, alternatively, marine protected areas; and iii) predictive modelling (Borja et al., 2012, 2013; Smith, 2016).

Historical data are rarely available, non-homogeneous, incomplete and seldom reliable, because of the lack of standardization in the sampling methods, changes in technology and observer effects (Leriche et al., 2004; Montefalcone et al., 2013; Gatti et al., 2015). When historical data are available, they may provide precious information to understand magnitude and pattern of change in the long term evolution of marine ecosystems (Canessa et al., 2017; Gatti et al., 2017). However, they are often only descriptive and go back in the past normally for a few decades only (Bianchi and Morri, 2004), thus in periods where a shift over time in the expectation of what a healthy ecosystem looks like can be observed - shifting or sliding baselines syndrome (Pauly, 1995). It means that, when change to a system is measured against previous available baselines that have already experienced significant alterations from the original state of the system, we might fail in defining what is really "natural" and then loss the perception of real change.

With few exceptions, pristine situations can be considered as definitively lost today (Jackson and Sala, 2001; Stachowitsch, 2003; Duarte et al., 2008), especially in coastal ecosystems that have been affected by significant impacts in the last decades, such as coral and rocky reefs and seagrass meadows (De'ath et al., 2012; Bianchi et al., 2014; Ponti et al., 2014; Montefalcone et al., 2015). Adopting marine protected areas as reference might be ineffective because protected habitats often showed the same status of unprotected ones, as observed in seagrass meadows (Montefalcone et al., 2009), infralittoral rocky reefs (Parravicini et al., 2013) and coralligenous reefs (Montefalcone et al., 2017).

Despite its practical and methodological limitations (Vacchi et al., 2014a), modelling remains in many situations the best approach with interesting potential still little explored (Parravicini et al., 2012; Vacchi et al., 2013). Predictive habitat modelling, from simple empirical models to detailed mechanistic and complex process-based approaches, have been recently developed to investigate the potential effects of physical environment on individual plants and to predict seagrass occurrence (Vacchi et al., 2010, 2013, 2014b; Detenbeck and Rego, 2015 and reference therein). Mechanistic approaches are based on a number of interactive physical parameters (e.g., wave energy, light, substrate typology, salinity, temperature) for which georeferenced data are available, and for which optima or thresholds for seagrass growth and survival can be obtained from the literature (Downie et al., 2013). More complex models also incorporated emergent properties influencing the growth, the loss rates, and the interactions between seagrass and their environment (Wortman et al., 1997; Fonseca et al., 2004; Kendrick et al., 2005).

The Mediterranean Sea is strongly affected by local and global impacts and experienced heavy alterations during the last two centuries (Bianchi et al., 2012), so that defining proper baselines for evaluating the rate of change in its ecosystems is compulsory. In this paper, we applied a predictive geospatial 2D model for the first time at the Mediterranean spatial scale on an ecosystem engineer seagrass species. The predictive modelling is based on beach morphodynamic features to define the reference conditions for the upper limit (i.e., the landward continuous front of the meadow) of seagrass meadows. Seagrass meadows are key coastal habitats and are used in monitoring plans as bioindicators of environmental health, thanks to their sensitivity to human-induced pressures (Pergent-Martini et al., 2005; Montefalcone, 2009; Boudouresque et al., 2012). In particular, the meadow upper limit is commonly used as an indicator of meadow health, being directly influenced by pressures coming from the coast (Pergent et al., 2005; Montefalcone, 2009; Boudouresque et al., 2012). Although seagrass meadows may be naturally fragmented by waves, currents, and colonization processes into patches of different size and form (Pace et al., 2017), wide-scale fragmentation of the meadow in correspondence of the upper limit has been shown to be a direct effect of high levels of coastal anthropization (Montefalcone et al., 2010).

*Posidonia oceanica*, the most important and abundant seagrass of the Mediterranean Sea (Boudouresque et al., 2012; Vacchi et al., 2017), forms extensive meadows that border most Mediterranean coasts (Telesca et al., 2015). Detailed maps of *P. oceanica* meadows were combined on a GIS platform with a nearshore hydrodynamic model (i.e., a model able to simulate the wave processes in the nearshore zone) to predict the theoretical natural position (i.e., the baseline) of the meadow upper limit according to the beach morphodynamics, i.e. the distinctive type of beach produced by local geomorphology and wave climate (Folkard, 2005; Jackson et al., 2005; Infantes et al., 2009). The predictive model, already tested at regional spatial scale along the Ligurian coast (NW Mediterranean Sea), showed perfect agreement between predictions and observations (Vacchi et al., 2010, 2014a, b). We thus extended the application of this predictive model at (western and central) Mediterranean spatial scale on *Posidonia oceanica* meadows along coastal areas showing different coastal morphologies and hydrodynamic characteristics, and affected by a number of natural and/or human local disturbances, to spatially predict the reference conditions of seagrass meadows against which to evaluate the change experienced by the upper limit of these priority ecosystems. We hypothesise that predictions can discriminate between sites subjected to different levels of coastal pressures: in areas with low pressures, the position of the meadow upper limit is expected to be found within the reference condition zone for the most of its extent and with no or little signs of regression (i.e., seaward withdrawal); in areas affected by high level of pressures, the upper limits is expected to lay deeper than the reference condition zone, and the seaward distance from this reference zone can be interpreted as the linear loss of meadow extent caused by anthropogenic activities and/or natural constrains.

## Materials and Methods

### Study areas

The predictive model was applied in eight coastal areas of Spain, France, Italy, and Greece (Fig. 1), harbouring important meadows of *Posidonia oceanica*: La Azohía (Murcia, eastern Spain); El Campello (Valencia, eastern Spain); Cavalaire-sur-Mer (Provence, south-eastern France); Saleccia (northern Corsica, France); Alassio (Liguria, northern Italy); Marina di Pescia Romana (Lazio, central Italy); Mondello (Sicily, southern Italy); and Acharavi (Corfu Island, north-western Greece). Morphodynamic characteristics (i.e., geomorphologic settings and wave exposures) of the eight areas are reported in Table 1. The main human and natural pressures affecting each costal area are summarized in Table 2. The level of pressures in each study site was evaluated through the use of the pressure level index (Piazzi et al., 2015, 2018). This index was defined as the sum of eight pressures affecting coastal areas (i.e., urbanization and urban waste, ports, tourism, industrial activities, agricultural waste, anchoring, sediment load by rivers, rip currents). Each pressure was classified from 0 (no pressure) to 2 (strong pressure), according to presence and type of the pressure, and to distance of the site from the pressure source (Table 2). The pressure level index ranges from a minimum value of 0 to 16.

**Application of the predictive model**

Application of the geospatial predictive 2D model requires the following three steps (Fig. 2): (i) characterization of the morphology and the depth of *Posidonia oceanica* meadow upper limit; (ii) definition of the near-shore morphodynamic domain (Jackson et al., 2005) and positioning of the breaking depth; and (iii) computation of the predictive equations to define the reference conditions (Vacchi et al., 2014b).

For step one, in each of the eight selected coastal areas, we combined high resolution (i.e., at the scale of 1:10000, but in El Campello, where the only available scale was 1:25000) thematic maps of *P. oceanica* meadows, resulted from single beam, multibeam and/or side scan sonar surveys (Table 4), with recent aerial imageries of the coastal zone (from Google Earth®) to define depth and morphology (continuous or fragmented) of the meadow upper limit (Fig. 3). In the selected areas, the real meadow upper limit was recognised as the front bordering the main body of the meadow (Bianchi and Peirano, 1995), where *P. oceanica* develops on either sand and other soft-bottoms or matte, the latter being an autogenic substrate consisting of interlaced remnants of roots, rhizomes and entangled sediment (Giovannetti et al., 2008). Stunted patches of *P. oceanica* established far in advance of the continuous front (Vacchi et al., 2010), and often settled on rocks, should not be considered as the real upper limit of the meadow but only as isolated outposts (Montefalcone et al., 2016).

For step two, we calculated the breaking depth ($d_b$), i.e. the depth where the wave breaks (Smith, 2003), which significantly controls the landward position of the meadow upper limit (Vacchi et al., 2010) and represents an essential variable in the prediction of the meadow upper limit position (Vacchi et al., 2017). We defined the annual breaking depth and its geographical position in each coastal area using the software MIKE21 SW, a third generation spectral wind-wave model based on unstructured meshes (Warren and Bach, 1992; Jose and Stone, 2006; Martinelli et al., 2006). The model simulates growth, decay and transformation of wind-generated waves and swells in offshore and coastal areas. The fully spectral formulation of MIKE 21 SW was used, which is based on the wave action balance equation (Hardy and Young, 1996; Komen et al., 1996). The bathymetric input of the model was derived for each study area from maps at the scale of 1:5000. Relevant local off-shore wave parameters were obtained from ondametric buoys and/or high resolution hydrodynamic studies (Table 1): offshore wave height ($H_0$), offshore wave length ($L_0$), and offshore wave period ($T_0$) (return time 1 year) (Fig. 3). According to Infantes et al. (2009) and Vacchi et al. (2014b), we employed annual offshore wave parameters (return time 1 year) instead of the simply daily average waves, as the latter could underestimate the effect of annual extreme events on the meadow. *Posidonia oceanica* is a long-lived species and the position of the lower limit of its meadows is therefore likely to be controlled more by extreme storm waves than by short period wave climate (Vacchi et al., 2010).

Finally, for step three, the predictive model elaborated by Vacchi et al. (2010, 2014a, b) was applied to the eight coastal areas. The model proposes two equations to locate the natural position of the *P. oceanica* meadow upper limit, i.e. identify a seafloor portion where the upper limit should lie due to hydrodynamics and in absence of major human pressures. The equations were derived from a linear model showing the best performance to predict the theoretical natural position of the meadow upper limit with respect to the breaking depth (Vacchi et al., 2014b). Uncertainty of model intercept and slope was estimated as 95% confidence interval of the parameters obtained by fitting the model on 500 bootstrap replicates of the original dataset. According to the 95% confidence interval of model parameters obtained by bootstrapping, two equations were identified:

i) $k_{min} = 5.94 + 0.29\varepsilon$

ii) $k_{max} = 17.83 + 0.41\varepsilon$

where $k_{min}$ and $k_{max}$ are two points along the submerged beach profile as predicted using respectively the 2.5% and the 97.5% of the parameters obtained by the bootstrapping procedure when modelling the difference of the breaking depth and the observed upper limit position as a function of the surf scaling index (ε) (Dean and Dalrymple, 2004). The former is the expected minimum linear distance (in meters) of the upper limit from the breaking depth, the latter is the expected maximum distance (in meters) of the upper limit from the breaking depth (see Fig. 2). The surf scaling index is computed as $\varepsilon = a\omega^2 / g\tan^2\beta$, where a (breaker amplitude) = $H_0/2$, ω (incident wave radian energy) = $2\pi/T_0$, g = acceleration of gravity; β = slope of the beach in the surf zone, $H_0$ = offshore wave height; $T_0$ = offshore wave period. Meadow upper limit is found very close to the breaking depth in those sites characterized by lower ε values, whereas higher ε values corresponded to larger distances (Vacchi et al., 2014b).

The spatially modelled reference conditions of the meadow upper limits, defined as the portions of the seafloor between the seaward ($k_{max}$) and the landward ($k_{min}$) boundaries computed by the model, were then positioned on the detailed thematic cartographies of *P. oceanica* meadows in a Geographical Information System (GIS) environment (Fig. 3). These portions were obtained with the buffer geo-processing tool available in ArcGis 10.3 software.

To evaluate the state of health of the meadow upper limit, three metrics have been used: the mean regressed distance, the proportion of regression, and the degree of habitat fragmentation. In each study area the mean regressed distance (i.e., the mean extent of the seaward withdrawal) of the meadow upper limit was computed on maps after running the model, averaging ten replicated measures (uniformly spaced along the upper limit) of the distance (in meter) between the position of the $k_{max}$ and the position of the observed upper limit. The proportion of the meadow upper limit showing regression (in %) was computed rating the total length of the healthy limit with the length of the limit located seaward from the reference condition zones (i.e., regressed). Inspiring to the approach of Montefalcone et al. (2010), the degree of habitat fragmentation has been computed as the percentage of meadow discontinuities in correspondence of its upper limit directly on maps. With the ArcGis drawing tool we contoured with a polyline the whole extent of the observed meadow upper limit on each map, and then we measured (in meters) the total length of the polyline containing *P. oceanica* (P) and the total length of the polyline not containing *P. oceanica*, i.e. laying either on dead matte or on other soft-bottoms (O). We used the following formula to compute the degree of habitat fragmentation: habitat fragmentation = (O/O+P) × 100.

Being computed directly on maps, the habitat fragmentation is an independent measure of the meadow health condition with respect to the other two metrics that derived from the application of the predictive modelling. We thus tested relationships among these three metrics using linear regressions on a total of 8 observations. Linear regressions were also used to test relationships between the pressure level index and the three metrics of the meadow state of health.

## Results

Variability of the breaking depth ($d_b$) among study areas is mainly influenced by differences in wave exposure and seafloor morphology (Table 5). Shallower breaking depth values were found in Mondello, whilst the deeper in Saleccia. All the beaches showed a highly dissipative morphodynamic domain, with the surf scaling index >100. The average values of $k_{min}$ ranged from 50±11 m in Cavalaire-sur-Mer to 500±379 m in Acharavi. The average values of $k_{max}$ ranged from 79±19 m in Cavalaire-sur-Mer to 716±536 m in Acharavi.

Significant relationships were found among the three metrics adopted to evaluate the meadow state of health, i.e. the mean regressed distance, the proportion of regression and the degree of habitat fragmentation (Fig. 4). The three metrics were also spatially correlated to the main human and natural pressures affecting each costal area, and their correlations with the pressure level index were significant (Fig. 4). In coastal areas with the lowest values of the index, the meadow upper limit showed better condition than in highly developed areas (Table 2 and Table 3).

From the maps of *Posidonia oceanica* meadows, all the upper limits appeared fragmented at the scale of analysis, and plotting the respective predicted reference condition zones (the seafloor region between $k_{min}$ and $k_{max}$) their regression resulted obvious (Fig. 5, Fig. 6, Table 3). In Acharavi, where the lowest value of the pressure level index has been obtained, the upper limit of the *P. oceanica* meadow laid mostly (82.3%) within the reference condition zone and appeared little fragmented and with a short extent of seaward withdrawal. In Alassio, Mondello, Cavalaire-sur-Mer, Marina di Pescia Romana, and Saleccia the meadow upper limits laid only partially within the predicted reference zone and the pressure level index displayed intermediate values. In Alassio the meadow upper limit was located at a mean distance of 10 m seaward from the lower boundary of the reference conditions zone, and showed highest degrees of fragmentation and regression mainly in its western sector. In Mondello, half the extent of the limit (55.4%) laid within the reference zone, but the wide sandy channel in the southern portion of the meadow resulting from rip currents implied a mean regressed distance of 87 m. In Cavalaire-sur-Mer the upper limit appeared highly fragmented throughout the whole sector investigated and remained within the reference condition zone for only 38.9%. In Marina di Pescia Romana meadow the upper limit was located at a mean distance of 40 m seaward from the reference zone, with the eastern sector being the most regressed. In Saleccia, only 36.1% of the upper limit laid inside the reference condition zone but was highly fragmented; the meadow was interrupted by a large natural sandy channel (an erosive structure related to rip currents), mean regressed distance being 100 m. In La Azohía and El Campello, which displayed the highest values of the pressure level index, the upper limits were almost entirely located seaward from the reference zone (87.4% and 98.9%, respectively) and appeared highly fragmented. In La Azohía, the regression of the upper limit was especially obvious in the central and eastern sectors of the meadow, with a mean regressed distance of 126 m with respect to the $k_{max}$ value. A mean regressed distance of 133 m was also evidenced in El Campello, notwithstanding the comparatively lower value of habitat fragmentation with respect to La Azohía.

## Discussion

The natural position of the *Posidonia oceanica* meadow upper limit can be predicted on the basis of physical parameters (Vacchi et al., 2010, 2014b, 2017). The geospatial 2D predictive model formerly employed to locate the reference condition of the meadow upper limits at regional spatial scale in Liguria, NW Mediterranean (Vacchi et al., 2014b; Burgos et al., 2017), was here exported and applied at the Mediterranean spatial scale on meadows under different disturbance regime, and showed effective to define proper baselines for assessing changes experienced by seagrass. When the model is applied on coastal areas affected by low levels of human and natural pressures, the position of the meadow upper limits is expected to be found within the reference condition zone and can be defined as the baseline to evaluate future change. In areas affected by strong human pressures, or under the influence of natural constrains (such as river inputs and rip currents), the meadow upper limit is expected to exhibit unhealthy conditions, appearing fragmented on maps and laying deeper than the reference (i.e., deeper than $k_{max}$). Thus, the linear loss of meadow extent seaward can thus be informative of the regression experienced by the meadow in time (Vacchi et al., 2014b).

Our study evidenced the regression of the meadow upper limit in all the localities investigated, with the least values in Acharavi and the greatest in El Campello, thus confirming the already reported wide decline of *P. oceanica* meadows in the Mediterranean Sea (Boudouresque et al., 2009; Marbà et al., 2014; Holon et al., 2015; Telesca et al., 2015; Burgos et al., 2017). This pattern is due to environmental alterations and physical damages and is mainly the result of the synergic effect of local and global impacts (Pergent et al., 2012; Giakoumi et al., 2015). Hydrodynamics, substrate typology and extreme climatic events, along with local human disturbances (e.g., beach nourishments, water turbidity, pollution, and anchoring) are all factors that could affect the condition of the meadow upper limits.

The comparatively healthier condition found in the meadow of Acharavi is consistent with the low level of coastal pressures (Malltezi et al., 2010; Prevenios et al., 2017). The higher values of all the three metrics found in Saleccia, another area little impacted by coastal urbanization, were likely due to natural factors, such as the input by the adjacent Liscu River and the presence of strong rip currents (Bonacorsi et al., 2013). Both factors hamper the development of *P. oceanica* meadows, because of decreased water salinity and creation of erosive structures (often called 'return rivers'), respectively (Ben Alaya, 1972; Boudouresque et al., 2012). In the remaining meadows from France, Spain, and Italy, the upper limits showed regressed and fragmented, in coincidence with the intense local anthropogenic coastal pressures. In Alassio and Mondello, for instance, the healthy portions of the meadow upper limits were slightly larger than the unhealthy ones: both meadows develop along highly touristic coastal areas but are comparatively far from industrial and large urban centres (Calvo et al., 1993; Montefalcone et al., 2009). On the contrary, the meadows of Cavalaire-sur-Mer and Marina di Pescia Romana had greater proportions of unhealthy than healthy upper limits. In Cavalaire-sur-Mer, the fragmented and regressed upper limit of the meadow was likely to be due to the influence of the nearby marina and the consequent intense mooring pressure of yachting activities, notwithstanding the occurrence of an authorized anchorage zone (Andromède Océanologie and Egis Eau, 2011). In Marina di Pescia Romana, the occurrence of an unhealthy upper limit only in the eastern sector of the meadow may be linked to the presence of a powerhouse that discharges waste waters into the sea directly over the meadow (Ardizzone et al., 2006). In La Azohía, waste waters coming from the numerous greenhouses located on land at a short distance from the coastline might be the cause of the highly fragmented and regressed upper limit. In El Campello, the meadow upper limit was almost completely regressed, being located close to the city of Alicante and was under the direct influence of its numerous human activities. Unhealthy conditions of these two meadows are consistent with the huge decline described for most seagrass meadows of Spain between 1967 and 1992 (Duarte, 1995; Marbà et al., 1996, 2002; Marbà, 2009).

The three metrics used to evaluate the state of health of the meadow upper limits effectively discriminated coastal areas according to their level of pressures and showed consistent results in all the meadows investigated. Habitat fragmentation is a metric strictly dependent by the scale of the maps, in particular by the spatial resolution and the extent (Wu, 2004). Except from El Campello, our maps displayed the same extent but showed different resolutions due to different detail in the original map digitalization: results of habitat fragmentation might be influenced by lower resolution in the least detailed maps.

Human pressures along the Mediterranean coasts are expected to dramatically increase in the next decades (Micheli et al., 2013); the consequent increase of water eutrophication, together with sea-level rise due to global warming, will further amplify seagrass decline if no specific management strategies are undertaken (Diaz-Almela et al., 2007; Marbà and Duarte, 2010; Gacía et al., 2012, 2013). The effect of the 20th century sea level rise acceleration in the Mediterranean (e.g., Vacchi et al., 2016) on the shallow portion of seagrass meadows is presently not fully understood. The significant increase of the rising rates in the next century (Church et al., 2013) is likely to affect the efficacy and accuracy of our predictions and should thus be taken into account in any future tuning of the model.

*Posidonia oceanica* might lose 75% of its suitable habitat by 2050, risking functional extinction by 2100 due to direct and indirect effects of climate change (Jorda et al., 2012; Chefaoui et al., 2018). On the other hand, along the Mediterranean French coast, upper limits of *P. oceanica* meadows showed mostly stable during the last 85 years, despite coastal development and increase of impacting human activities (Holon et al., 2015). The maintenance (or restoration) of the "good environmental status" by 2020 imposed by the Marine Strategy Framework Directive urgently asks for the identification of the reference conditions against which the current state of our ecosystems must be compared. Availability of such a predictive tool to define baselines when other approaches (i.e., historical data and pristine areas) might be absent or inadequate (Bonacorsi et al., 2013), is essential in the frame of an integrated management plan of seagrass meadows at the Mediterranean spatial scale. Also, to effectively slow down and possibly reverse this decline, targeted local conservation measures must be identified to reduce human pressure and mitigate effects of global change (Orth et al., 2006), as well as specific interventions of transplantation in degraded meadows (Pirotta et al., 2015). Our predictive model might help identifying potential areas where conservation and management actions must be concentrated. Identification of suitable portions of the bottom where seagrass can be effectively restored is another potential application of our model, provided that the pressures that caused regression are removed.

Differently from other mechanistic models already developed to predict distribution of seagrass meadows (Peirano and Bianchi, 1997; Detenbeck and Rego, 2015 and references therein), our predictive model has a relatively easy formulation being mainly dependent on a single physical parameter (i.e., the breaking depth). The accurate positioning of $k_{min}$ and $k_{max}$ parameters is strongly influenced by the occurrence of artificial hard structures along the coast, because they modify wave motion altering the local hydrodynamics. Nevertheless, the model may be applied where the coastline has been largely modified, provided that the assessment of the original terrain morphology remains possible (Burgos et al., 2017). Clearly there is also a need of detailed maps of *P. oceanica* meadows (at least of their shallow portions) and detailed bathymetries upon which to build the model. The scale used on maps is another important aspect to define the reference condition zone: maps with a scale ≥ 1:25000 are mandatory to obtain reliable results and improve interpretation. This is the reason why the coastal areas we selected to apply the model are all located in the western and the central Mediterranean Sea, because of the little availability of suitable data on seagrass meadows in the southern and the eastern regions of the Mediterranean Sea. Future application of our methodology in other areas of the Mediterranean Sea would include beaches with intermediate and reflective morphodynamic domains. Moreover, exporting the model in other geomorphologic, climatic and hydrodynamic contexts may lead to important results in the assessment of long-time changes experienced by seagrass ecosystems due to global and local human impacts.

As already underlined by previous studies (Vacchi et al., 2010, 2014b), our model can effectively predict the natural landward distribution of meadows settled on sand or other soft bottoms and on matte. *P. oceanica* is also able to grow in small and discontinuous stands landward from the breaking depth and far in advance from the continuous meadow front. This may especially occur on rocks, where the rhizomes of *P. oceanica* are able to anchor strongly (Montefalcone et al., 2016), thus allowing the plant to withstand the intense hydrodynamics that characterize the surf zone (Vacchi et al., 2017). These shallow patches are of great interest, and should therefore be monitored, as they could coalesce to originate a new continuous upper limit of the meadow under the future scenarios of sea level rise (Pergent et al., 2014).

The canopy of dense meadows is known to attenuate waves (Peterson et al., 2004; Pace et al., 2017), which is likely to cause a landward shift of the breaking depth. We had no measurements of wave attenuation provided by the canopies, and our model took into account off shore waves. Thus, the observed regression of the meadow upper limit may be even greater than what we have been able to measure. In this sense, our measures are likely to be very conservative. As in any case of model application, field surveys, possibly with underwater inspections, must be carried out to sea-truth and validate the results of the model.

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## References

Andromède Océanologie, Egis Eau. 2011. Volet environnemental du schéma départemental de la mer et du littoral: Etat des lieux. Conseil Général du Var, 252 pp.

Ardizzone G., Belluscio A., Maiorana L. 2006. Long-term change in the structure of *Posidonia oceanica* landscape and its reference for a monitoring plan. Marine Ecology 27, 299-309.

Ben Alaya H. 1972. Répartition et conditions d'installation de *Posidonia oceanica* Delile et *Cymodocea nodosa* Ascherson dans le golfe de Tunis. Bulletin de la Station Océanographique de Salammbô 2 (3), 331-416.

Benoit G., Comeau A. 2005. A sustainable future for the Mediterranean. The Blue Plan's environment and development outlook. UK: Earthscan London, 450 pp.

Bianchi C.N., Corsini-Foka M., Morri C., Zenetos A. 2014. Thirty years after: dramatic change in the coastal marine ecosystems of Kos Island (Greece), 1981-2013. Mediterranean Marine Science 15 (3), 482-497.

Bianchi C.N., Morri C. 2004. Climate change and biological response in Mediterranean Sea ecosystems – a need for broad-scale and long-term research. Ocean Challenge 13 (2), 32-36.

Bianchi C.N., Morri C., Chiantore M., Montefalcone M., Parravicini V., Rovere A. 2012. Mediterranean Sea biodiversity between the legacy from the past and a future of change. In: Stambler N. (Ed.), Life in the Mediterranean Sea: A Look at Habitat Changes. New York: Nova Science, 55 pp.

Bianchi C.N., Peirano A. 1995. Atlante delle fanerogame marine della Liguria: *Posidonia oceanica* e *Cymodocea nodosa*. ENEA, Centro Ricerche Ambiente Marino. La Spezia, 146 pp.

Bonacorsi M. 2012. Caractérisation des peuplements benthique du Cap Corse. PhD Thesis. Université de Corse Pascal Paoli, Corte, France, 165 pp.

Bonacorsi M., Pergent-Martini C., Bréand N., Pergent G. 2013. Is *Posidonia oceanica* regression a general feature in the Mediterranean Sea? Mediterranean Marine Science 14, 193-203.

Borja Á., Dauer D.M., Grémare A. 2012. The importance of targets and reference conditions in assessing marine ecosystem quality. Ecological Indicators 12, 1-7.

Borja Á., Elliott M., Andersen J.H., Cardoso A.C., Carstensen J., Ferreira J.G., Heiskanen A.S., Marques J.C., Neto J.M., Teixeira H., Uusitalo L., Uyarra M.C., Zampoukas N. 2013. Good Environmental Status of marine ecosystems: what is it and how do we know when we have attained it? Marine Pollution Bulletin 76, 16-27.

Borja Á., Elliott M., Carstensen J., Heiskane, A.S., van de Bund W. 2010. Marine management - Towards an integrated implementation of the European Marine Strategy Framework and the Water Framework Directives. Marine Pollution Bulletin 60, 2175-2186.

Boudouresque C.F., Bernard G., Bonhomme P., Charbonnel E., Diviacco G., Meinesz A., Pergent G., Pergent-Martini C., Ruitton S., Tunesi L. 2012. Protection and conservation of *Posidonia oceanica* meadows. Monaco: RaMoGe Publication, 202 pp.

Boudouresque C.F., Bernard G., Pergent G., Shili A., Verlaque M., 2009. Regression of Mediterranean seagrasses caused by natural processes and anthropogenic disturbances and stress: a critical review. Botanica Marina 52, 395-418.

Burgos E., Montefalcone M., Ferrari M., Paoli C., Vassallo P., Morri C., Bianchi C.N. 2017. Ecosystem functions and economic wealth: trajectories of change in seagrass meadows. Journal of Cleaning Production 168, 1108-1119.

Frada Orestano C., Abbadessa P. 1993. The suitability of a topographical instrument for an integrated approach to the cartography of *Posidonia oceanica* meadows. Oceanologica Acta 16 (3), 273-278.

Canessa M., Montefalcone M., Bavestrello G., Povero P., Coppo S., Morri C., Bianchi C.N. 2017. Fishery maps contain approximate but useful information for inferring the distribution of marine habitats of conservation interest. Estuarine, Coastal and Shelf Science 187, 74-83.

Chefaoui R.M., Duarte C.M., Serrão E.A. 2018. Dramatic loss of seagrass habitat under projected climate change in the Mediterranean Sea. Global Change Biology, in press.

Church J.A., Clark P.U., Cazenave A., Gregory J.M., Jevrejeva S., Levermann A., Merrield M.A., Milne G.A., Nerem R.S., Nunn P.D., Payne A.J., Pfeffer W.T., Stammer D., Unnikrishnan A.S. 2013. Sea level change. In: Stocker T.F., Qin D., Plattner G.-K., Tignor M., Allen S.K., Boschung J., Nauels A., Xia Y., Bex V., Midgley P.M. (Eds.). Climate change 2013: The physical science basis. Contribution of Working Group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

de Andrés M., Barragán J.M. 2016. Urban-coastal development. Study method for quantifying in a global scale. Revista de Estudios Andaluces 33 (1), 64-83.

Dean R.G., Dalrymple R.A. 2004. Coastal processes with engineering applications. UK: Cambridge University Press, 488 pp.

De'ath G., Fabricius K.E., Sweatman H., Puotinen M. 2012. The 27-year decline of coral cover on the Great Barrier Reef and its causes. Proceeding of the National Academy of Science of the USA 109 (44), 17995-17999.

Detenbeck N.E., Rego S. 2015. Predictive seagrass habitat model. EPA/600/R-15/003. U.S. Environmental Protection Agency, Narragansett, RI, 166 pp.

Diaz-Almela E., Marbà N., Duarte C.M. 2007. Consequences of Mediterranean warming events in seagrass (*Posidonia oceanica*) flowering records. Global Change Biology 13, 224-235.

Diviacco G., Coppo S. 2007. Atlante degli habitat marini della Liguria: descrizione e cartografia delle praterie di *Posidonia oceanica* e dei principali popolamenti marini costieri. Genova: Regione Liguria, 205 pp.

Diviacco G., Spada E., Virno Lamberti C. 2001. Le fanerogame del Lazio: descrizione e cartografia delle praterie di *Posidonia oceanica* e dei prati di *Cymodocea nodosa*. Roma: ICRAM, 113 pp.

Downie A-L., von Numers M., Boström C. 2013. Influence of model selection on the predicted distribution of the seagrass *Zostera marina*. Estuarine, Coastal and Shelf Science 121/122, 8-19.

Duarte C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41, 87-112.

Duarte C.M., Conley D.J., Carstensen J., Sánchez-Camacho M. 2008. Return to Neverland: shifting baselines affect eutrophication restoration targets. Estuaries and Coasts 32 (1), 29-36.

European Council 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 on establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Communities L 164, 19-40.

Fernández-Torquemada Y., Díaz-Valdés M., Colilla F., Luna B., Sánchez-Lizaso J.L., Ramos-Esplá A.A. 2008. Descriptors from *Posidonia oceanica* (L.) Delile meadows in coastal waters of Valencia, Spain, in the context of the EU Water Framework Directive. ICES Journal of Marine Science 65, 1492-1497.

Folkard A.M. 2005. Hydrodynamics of model *Posidonia oceanica* patches in shallow water. Limnology and Oceanography 50, 1592-1600.

Fonseca M.S., Whitfield P.E., Kenworthy W.J., Colby D.R., Julius B.E. 2004. Use of two spatially explicit models to determine the effect of injury geometry on natural resource recovery. Aquatic Conservation: Marine and Freshwater Ecosystems 14, 281-298.

Gacía R., Sánchez-Camacho M., Duarte C.M., Marbà N. 2012. Warming enhances sulphide stress of Mediterranean seagrass (*Posidonia oceanica*). Estuarine, Coastal and Shelf Science 113, 240-247.

Gacía R., Holmer M., Duarte C.M., Marbà N. 2013. Global warming enhances sulphide stress in a key seagrass species (NW Mediterranean). Global Change Biology 19, 3629-3639.

Gatti G., Bianchi C.N., Montefalcone M., Venturini S., Diviacco G., Morri C. 2017. Observational information on a temperate reef community helps understanding the marine climate and ecosystem shift of the 1980-90s. Marine Pollution Bulletin 114, 528-538.

Gatti G., Bianchi C.N., Parravicini V., Rovere A., Peirano A., Montefalcone M., Massa F., Morri C. 2015. Ecological change, sliding baselines and the importance of historical data: lessons from combining observational and quantitative data on a temperate reef ove 70 years. PLoS One 10 (2), e0118581.

Giakoumi S., Halpern B.J., Michel L., Gobert S., Sini M., Boudouresque C.F., Gambi M.C., Katsanevakis S., Lejeune P., Montefalcone M., Pergent G., Pergent-Martini C., Sanchez-Jerez P., Velimirov B., Vizzini S., Abadie A., Coll M., Guidetti P., Micheli F., Possingham H.P. 2015. Developing a framework for assessment of cumulative human impacts on food webs and management action prioritization. Conservation Biology 29 (4), 1228-1234.

Giovannetti E., Lasagna R., Montefalcone M., Bianchi C.N., Albertelli G., Morri C. 2008. Inconsistent responses to substratum nature in *Posidonia oceanica* meadows: an integration through complexity levels? Chemistry and Ecology 24 (S1), 145-153.

Halpern B.S., Frazier M., Potapenko J., Casey K.S., Koenig K., Longo C., Lowndes J.S., Rockwood R.C., Selig E.R., Selkoe K.A., Walbridge S. 2015. Spatial and temporal changes in cumulative human impacts on the world’s ocean. Nature Communications 6, 7615.

Hardy T.A., Young I.R. 1996. Field study of wave attenuation on an offshore coral reef. Journal of Geophysical Research 101 (C6), 311-326.

Holon F., Boissery P., Guilbert A., Freschet E., Deter J. 2015. The impact of 85 years of coastal development on shallow seagrass beds (*Posidonia oceanica* L. (Delile)) in South Eastern France: a slow but steady loss without recovery. Estuarine, Coastal and Shelf Science 165, 204-212.

Infantes E., Terrados J., Orfila A., Cañellas B.Á., lvarez-Ellacuria A. 2009. Wave energy and the upper depth limit distribution of *Posidonia oceanica*. Botanica Marina 52, 419-427.

Jackson D.W.T., Cooper J.A.G., Del Rio L. 2005. Geological control of beach morphodynamic state. Marine Geology 216, 297-314.

Jackson J.B.C., Sala E. 2001 Unnatural oceans. Scientia Marina 65, 273-281.

Jorda G., Marbà N., Duarte C.M. 2012. Mediterranean seagrass vulnerable to regional climate warming. Nature 2, 821-824.

Jose F., Stone G.W. 2006. Forecast of nearshore wave parameters using MIKE-21 spectral wave model. Gulf Coast Association of Geological Societies Transactions 56, 323-327.

Kendrick G.A., Duarte C.M., Marba N. 2005. Clonality in seagrasses, emergent properties and seagrass landscapes. Marine Ecology Progress Series 290, 291-296.

Komen G.J., Cavaleri L., Donelan M., Hasselmann K., Hasselmann S., Janssen P.A.E.M. 1996. Dynamics and modelling of ocean waves. New York: Cambridge University Press, 532 pp.

Leriche A., Boudouresque C.F., Bernard G., Bonhomme P., Denis J. 2004. A one-century suite of seagrass bed maps: can we trust ancient maps? Estuarine, Coastal and Shelf Science 59, 353-362.

Malltezi J., Bani A., Dini V., Papakroni H., Shehu A., Sulce S. 2010. Spatial and temporal trends of water quality at Saranda Bay, Albania. Fresenius Environmental Bulletin 19 (9), 1822-1832.

Marbà N. 2009. Loss of seagrass meadows from the Spanish coasts: results of the Praderas project. In: Duarte C.M. (Ed.), Global loss of coastal habitats: rates, causes, and consequences. Bilbao: Fundación BBVA, 184 pp.

Marbà N., Díaz-Almela E., Duarte C.M. 2014. Mediterranean seagrass (*Posidonia oceanica*) loss between 1842 and 2009. Biological Conservation 176, 183-190.

Marbà N., Duarte C.M. 2010. Mediterranean warming triggers seagrass (*Posidonia oceanica*) shoot mortality. Global Change Biology 16, 2366-2375.

Marbà N., Duarte C.M., Cebrián J., Enríquez S., Gallegos M.E., Olesen B., Sand-Jensen K. 1996. Growth and population dynamics of *Posidonia oceanica* on the Spanish Mediterranean coast: elucidating seagrass decline. Marine Ecology Progress Series 137, 203-213.

Marbà N., Duarte C.M., Holmer M., Martínez R., Basterretxea G., Orfila A. Jordi A., Tintoré J. 2002. Effectiveness of protection of seagrass (*Posidonia oceanica*) populations in Cabrera National Park (Spain). Environmental Conservation 29 (4), 509-518.

Martinelli L., Zanuttigh B., Lamberti A. 2006. Hydrodynamic and morphodynamic response of isolated and multiple low crested structures: experiments and simulations. Coastal Engineering 53 (4), 363-379.

Micheli F., Halpern B.S., Walbridge S., Ciriaco S., Ferretti F., Fraschetti S., Lewison R., Nykjaer L., Rosenberg A.A. 2013. Cumulative human impacts on Mediterranean and Black Sea marine ecosystems: assessing current pressures and opportunities. PLoS One 8 (12), e79889.

Montefalcone M. 2009. Ecosystem health assessment using the seagrass *Posidonia oceanica*: a review. Ecological Indicators 9, 595-604.

Montefalcone M., Albertelli G., Morri C., Parravicini V., Bianchi C.N. 2009. Legal protection is not enough: *Posidonia oceanica* meadows in marine protected areas are not healthier than those in unprotected areas of the northwest Mediterranean Sea. Marine Pollution Bulletin 58, 515-519.

Montefalcone M., Morri C., Bianchi C.N., Bavestrello G., Piazzi L. 2017. The two facets of species sensitivity: stress and disturbance on coralligenous assemblages in space and time. Marine Pollution Bulletin 117, 229-238.

Montefalcone M., Parravicini V., Vacchi M., Albertelli G., Ferrari M., Morri C., Bianchi C.N. 2010. Human influence on seagrass habitat fragmentation in NW Mediterranean Sea. Estuarine, Coastal and Shelf Science 86, 292-298.

Montefalcone M., Rovere A., Parravicini V., Albertelli G., Morri C., Bianchi C.N. 2013. Evaluating change in seagrass meadows: a time-framed comparison of Side Scan Sonar maps. Aquatic Botany 104, 204-212.

Montefalcone M., Vacchi M., Carbone C., Cabella R., Schiaffino C.F., Elter F.M., Morri C., Bianchi C.N., Ferrari M. 2016. Seagrass on the rocks: *Posidonia oceanica* settled on shallow-water hard substrata withstands wave stress beyond predictions. Estuarine, Coastal and Shelf Science 180, 114-122.

Montefalcone M., Vassallo P., Parravicini V., Paoli C., Morri C., Bianchi C.N. 2015. The exergy of a phase shift: ecosystem functioning loss in seagrass meadows of the Mediterranean Sea. Estuarine, Coastal and Shelf Science 156, 186-194.

Orth R.J., Carruthers T.J.B., Dennison W.C., Duarte C.M., Fourqurean J.W., Heck K.L., Hughes A.R., Kendrick G.A., Kenworthy W.J., Olyarnik S., Short F.T., Waycott M., Williams S.L. 2006. A global crisis for seagrass ecosystems. Bioscience 56 (12), 987-996.

Pace M., Borg J.A., Galdies C., Malhotra A. 2017. Influence of wave climate on architecture and landscape characteristics of *Posidonia oceanica* meadows. Marine Ecology 38, e12387.

Papathanassiou E., Zenetos A. 2005. State of Hellenic marine environment. Attiki, Greece: Hellenic Center for Marine Research Publ., 60 pp.

Parravicini V., Micheli F., Montefalcone M., Morri C., Villa E., Castellano M., Povero P., Bianchi C.N. 2013. Conserving biodiversity in a human-dominated world: degradation of marine sessile communities within a protected area with conflicting human uses. PLoS One 8 (10), e75767.

Parravicini V., Rovere A., Vassallo P., Micheli F., Montefalcone M., Morri C., Paoli C., Albertelli G., Fabiano M., Bianchi C.N. 2012. Understanding relationships between conflicting human uses and coastal ecosystems status: a geospatial modeling approach. Ecological Indicators 19, 253-263.

Pauly D. 1995. Anecdotes and the shifting baseline syndrome. Trends in Ecology & Evolution 10, 430.

Peirano A., Bianchi C.N. 1997. Decline of the seagrass *Posidonia oceanica* in response to environmental disturbance: a simulation-like approach off Liguria (NW Mediterranean Sea). In: Hawkins L.E., Hutchinson S. (Eds), The response of marine organisms to their environments. University of Southampton, UK, 87-95.

Pergent G., Bazairi H., Bianchi C.N., Boudouresque C.F., Buia M.C., Clabaut P., Harmelin-Vivien M., Mateo M.A., Montefalcone M., Morri C., Orfanidis S., Pergent-Martini C., Semroud R., Serrano O., Verlaque M. 2012. Mediterranean seagrass meadows: resilience and contribution to climate change mitigation. A short summary. Gland, Switzerland, and Málaga, Spain: IUCN, 40 pp.

Pergent G., Hocein B., Bianchi C.N., Boudouresque C.F., Buia M.C., Clabaut P., Harmelin-Vivien M., Mateo M.A., Montefalcone M., Morri C., Orfanidis S., Pergent-Martini C., Semroud R., Serrano O., Verlaque M. 2014. Climate change and Mediterranean seagrass meadows: a synopsis for environmental managers. Mediterranean Marine Science 15 (2), 462-473.

Pergent-Martini C., Leoni V., Pasqualini V., Ardizzone G.D., Balestri E., Bedini R., Belluscio A., Belsher T., Borg J., Boudouresque C.F., Boumaza S., Bouquegneau J.M., Buia M.C., Calvo S., Cebrian J., Charbonnel E., Cinelli F., Cossu A., Di Maida G., Dural B., Francour P., Gobert S., Lepoint G., Meinesz A., Molenaar H., Mansour H.M., Panayotidis P., Peirano A., Pergent G., Piazzi L., Pirrotta M., Relini G., Romero J., Sanchez-Lizaso J.L., Semroud R., Shembri P., Shili A., Tomasello A., Velimirov B. 2005. Descriptors of *Posidonia oceanica* meadows: use and application. Ecological Indicators 5, 213-230.

Personnic S., Boudouresque C.F., Astruch P., Ballesteros E., Blouet S., Bellan-Santini D., Bonhomme P., Thibault-Botha D., Feunteun E., Harmelin-Vivien M., Pergent G., Pergent-Martini C., Pastor J., Poggiale J.C., Renaud F., Thibaut T., Ruitton S. 2014. An ecosystem-based approach to assess the status of a Mediterranean ecosystem, the *Posidonia oceanica* seagrass meadow. PLoS One 9 (6), e98994.

Peterson C.H., Luettich Jr. R.A., Micheli F., Skilleter G.A. 2004. Attenuation of water flow inside seagrass canopies of differing structure. Marine Ecology Progress Series 268, 81-92.

Piazzi L., Gennaro P., Cecchi E., Serena F. 2015. Improvement of the ESCA index for the evaluation of ecological quality of coralligenous habitat under the European framework directives. Mediterranean Marine Science 16, 419-426

Piazzi L., Bianchi C.N., Cecchi E., Gatti G., Guala I., Morri C., Sartoretto S., Serena F., Montefalcone M. 2017. What's in an index? Comparing the ecological information provided by two indices to assess the status of coralligenous reefs in the NW Mediterranean Sea. Aquatic Conservation: Marine and Freshwater Ecosystems 27, 1091-1100.

Pirrotta M., Tomasello A., Scannavino A., Di Maida G., Luzzu F., Bellissimo G., Bellavia C., Costantini C., Orestano C., Sclafani G., Calvo S. 2015. Transplantation assessment of degraded *Posidonia oceanica* habitats: site selection and long-term monitoring. Mediterranean Marine Science 16 (3), 591-604.

Ponti M., Perlini R.A., Ventra V., Grech D., Abbiati M., Cerrano C. 2014. Ecological shifts in Mediterranean coralligenous assemblages related to gorgonian forest loss. PLoS One 9 (7), e102782.

Prevenios M., Zeri C., Tsangaris C., Liubartseva S., Fakiris E., Papatheodorou G. 2017. Beach litter dynamics on Mediterranean coasts: distinguishing sources and pathways. Marine Pollution Bulletin, http://dx.doi.org/10.1016/j.marpolbul.2017.10.013.

Rastorgueff P.A., Bellan-Santini D., Bianchi C.N., Bussotti S., Chevaldonné P., Guidetti P., Harmelin J.G., Montefalcone M., Morri C., Perez T., Ruitton S., Vacelet J., Personnic S. 2015. An ecosystem-based approach to evaluate the ecological quality of Mediterranean undersea caves. Ecological Indicators 54, 137-152.

Smith M.J. 2003. Surf zone hydrodynamics. In: Demirbilek Z. (Ed.), Coastal Engineering Manual, Part II, Coastal Hydrodynamics chapter II-4, Engineer Manual, 1110-2-1100. US Army corps of Engineers, Washington, DC.

Smith C.J., Papadopoulou K.N., Barnard S., Mazik K., Elliott M., Patrício J., Solaun O., Little S., Bhatia N., Borja A. 2016. Managing the marine environment, conceptual models and assessment considerations for the European Marine Strategy Framework Directive. Frontiers in Marine Science 3, 144.

Shochat E., Warren P.S., Faeth S.H., McIntyre N.E., Hope D. 2006. From patterns to emerging processes in mechanistic urban ecology. Trends in Ecology and Evolution 21, 186-191.

Stachowitsch M. 2003. Research on intact marine ecosystems: a lost era. Marine Pollution Bulletin 46 (7), 801-805.

Telesca L., Belluscio A., Criscoli A., Ardizzone G., Apostolaki E.T., Fraschetti S., Alagna A. 2015. Seagrass meadows (*Posidonia oceanica*) distribution and trajectories of change. Scientific Reports 5, 12505.

Vacchi M., De Falco G., Simeone S., Montefalcone M., Bianchi C.N., Morri C., Ferrari M. 2017. Biogeomorphology of the Mediterranean *Posidonia oceanica* meadows. Earth Surface Processes and Landforms 42, 42-54.

Vacchi M., Marriner N., Morhange C., Spada G., Fontana A., Rovere A. 2016. Multiproxy assessment of Holocene relative sea-level changes in the western Mediterranean: Sea-level variability and improvements in the definition of the isostatic signal. Earth-Science Reviews 155, 172-197.

Vacchi M., Misson G., Montefalcone M., Archetti R., Bianchi C.N., Ferrari M. 2013. Modelling reference conditions for the upper limit of *Posidonia oceanica* meadows. Rapports de la Commission Internationale pour l'Exploration Scientifique de la Mer Méditerranée 40, 579.

Vacchi M., Montefalcone M., Bianchi C.N., Morri C., Ferrari M. 2010. The influence of coastal dynamics on the upper limit of the *Posidonia oceanica* meadow. Marine Ecology 31, 546-554.

Vacchi M., Montefalcone M., Parravicini V., Rovere A., Vassallo P., Ferrari M., Morri C., Bianchi C.N. 2014a. Spatial models to support the management of coastal marine ecosystems: a short review of best practices in Liguria, Italy. Mediterranean Marine Science 15 (1), 189-197.

Vacchi M., Montefalcone M., Schiaffino C., Parravicini V., Bianchi C. N., Morri C., Ferrari M. 2014b. Towards a predictive model to assess the natural position of the *Posidonia oceanica* seagrass meadow upper limit. Marine Pollution Bulletin 83 (2), 458-466.

Warren I.R., Bach H. 1992. MIKE 21: a modeling system for estuaries, coastal waters and seas. Environmental Software 7 (4), 229-240.

Wortmann J., Hearne J.W., Adams J.B. 1997. A mathematical model of an estuarine seagrass. Ecological Modeling 98, 137-149.

Wu J. 2004. Effects of changing scale on landscape pattern analysis: scaling relations. Landscape Ecology 19, 125-138.

www.ifremer.fr/medar

www.mareografico.it

**Caption to figures and table headings**

Figure 1. Location of the eight study areas in the western and central basins of the Mediterranean Sea. La Azohía (Murcia, Spain), El Campello (Valencia, Spain), Cavalaire-sur-Mer (Provence, France), Saleccia (Corsica, France), Alassio (Liguria, Italy), Marina di Pescia Romana (Lazio, Italy), Mondello (Sicily, Italy), Acharavi (Corfu Island, Greece). Grid datum: D_WGS_1984 UTM zone 32 Northern Hemisphere.

Figure 2. A schematic representation of the steps required for the application of the geospatial model to predict the position of *Posidonia oceanica* meadow upper limit under natural conditions, viewed in vertical section: a) positioning of the present upper limit (dotted line), based on field data and/or available cartographies, and positioning of the breaking depth ($d_b$, black dot), based on nearshore hydrodynamics; b) positioning of $k_{min}$ and $k_{max}$, where the natural upper limit is expected to lie, as computed through the predictive equations described in the text.

Figure 3. A planimetric rendering in GIS environment of the procedure to predict the position of the *Posidonia oceanica* meadow upper limit under natural conditions, as illustrated in Figure 2. The orange portion of the seafloor represents the reference condition zone for meadow upper limit, as defined by $k_{min}$ and $k_{max}$ distances from the breaking depth.

Figure 4. Relationships between: (a) the regressed proportion and the mean regressed distance of the meadow upper limit; (b) the fragmentation and the mean regressed distance; (c) the fragmentation and the regressed proportion; (d) the pressure level index and the proportion of regression; (e) the pressure level index and the mean regressed distance; and (f) the pressure level index and the degree of habitat fragmentation. AZ = La Azohía; EC = El Campello;

CM = Cavalaire-sur-Mer; SA = Saleccia; AL = Alassio; MP = Marina di Pescia Romana; MO = Mondello; AC = Acharavi.

Figure 5. Detailed cartographies of *Posidonia oceanica* meadows in Spain (La Azohía, El Campello) and France (Cavalaire-sur-Mer, Saleccia), with superimposed the reference condition zones, which boundaries are defined by $k_{min}$ (yellow line) and $k_{max}$ (red line). The blue arrow at Saleccia indicates the main rip current path. Grid datum: D_WGS_1984 UTM zone 32 Northern Hemisphere.

Figure 6. Detailed cartographies of *Posidonia oceanica* meadows in Italy (Alassio, Marina di Pescia Romana, Mondello) and Greece (Acharavi), with superimposed the reference condition zones, which boundaries are defined by $k_{min}$ (yellow line) and $k_{max}$ (red line). The blue arrow at Mondello indicates the main rip current path. Grid datum: D_WGS_1984 UTM zone 32 Northern Hemisphere.

Table 1. Summary of the morphodynamic characteristics of the eight study areas and local off-shore wave parameters (return time 1 year). $H_0$ is the offshore wave height, $T_0$ is the offshore wave period and $L_0$ is the offshore wave length. Refer to Table 3 for bibliographic sources used in each area.

Table 2. Main natural and anthropogenic pressures affecting each costal area together with their relative score (from 0 to 2) and the value of the pressure level index.

Table 3. Mean values (± s.d.) of the depth of the meadow upper limits and the three metrics used to evaluate the healthy condition of the limit in each costal area: mean (± s.e.) regressed distance from $k_{max}$ value (in meters), regressed proportion (%), and degree of habitat fragmentation (%).

Table 4. Survey methods and scale of the thematic cartographies of *Posidonia oceanica* meadows used to characterize the morphology of the meadow upper limits in the eight study areas. Data on meadows characteristics, as well as on morphodynamics (see Table 1), were obtained by the following bibliographic sources: (1) www.ifremer.fr/medar; (2) Fernández-Torquemada et al. (2008); (3) Andromède Océanologie and Egis Eau (2011); (4) Bonacorsi (2012); (5) Diviacco and Coppo (2007); (6) Diviacco et al. (2001); (7) www.mareografico.it; (8) Papathanassiou and Zenetos (2005).

Table 5. Mean values (± s.d.) of the surf scaling index (ε), mean depth (± s.d.) of the breaking depth ($d_b$), and mean values (± s.d.) of the $k_{min}$ and the $k_{max}$ (expressed as the minimum and the maximum distance from the breaking depth, respectively) for each study area.

Table 1.

| **Locality** | **Beach orientation** | **Beach length (km)** | **Dominant wave** | **Beach slope (%)** | **$H_0$ (m)** | **$T_0$ (s)** | **$L_0$ (m)** |
|---|---|---|---|---|---|---|---|
| La Azohía (Murcia, Spain) | NW-SE | 5 | SO | 1.9 | 4.2 | 7.5 | 87.8 |
| El Campello (Valencia, Spain) | NE-SW | 6.5 | SE-NW | 1.5 | 5.0 | 9.0 | 126.7 |
| Cavalaire-sur-Mer (Provence, France) | NE-SW | 3 | S | 3.2 | 3.5 | 7.0 | 76.4 |
| Saleccia (Corsica, France) | E-O | 1 | NE | 3.8 | 5.5 | 9.1 | 129.2 |
| Alassio (Liguria, Italy) | NE-SW | 2 | SE | 3.0 | 2.6 | 5.8 | 52.4 |
| Marina di Pescia Romana (Lazio, Italy) | NW-SE | 10 | SE | 1.5 | 5.0 | 8.7 | 117.3 |
| Mondello (Sicily, Italy) | NW-SE | 2 | E | 2.0 | 2.5 | 9.0 | 126.4 |
| Acharavi (Corfu Island, Greece) | NE-SW | 3 | N | 1.0 | 4.1 | 7.7 | 76.3 |

Table 2.

| Locality | Urbanization and urban waste | Industrial activity | Ports | Tourism | Sediment load | Agricultural waste | Anchoring | Rip currents | Total |
|---|---|---|---|---|---|---|---|---|---|
| La Azohía (Murcia, Spain) | 1 | 1 | 1 | 2 | | 2 | 1 | | 8 |
| El Campello (Valencia, Spain) | 2 | 1 | 2 | 2 | | | 2 | | 9 |
| Cavalaire-sur-Mer (Provence, France) | 2 | | 1 | 2 | | | 2 | | 7 |
| Saleccia (Corsica, France) | | | | 1 | 2 | | 1 | 2 | 6 |
| Alassio (Liguria, Italy) | 1 | | 1 | 2 | | | 1 | | 5 |
| Marina di Pescia Romana (Lazio, Italy) | 1 | 2 | | 1 | | 2 | | | 6 |
| Mondello (Sicily, Italy) | 1 | | 1 | 2 | | | | 2 | 6 |
| Acharavi (Corfu Island, Greece) | 1 | | | 1 | | 1 | | | 3 |

Table 3.

| Locality | Depth (m) | Regressed distance (m) | Regressed proportion (%) | Habitat fragmentation (%) |
|---|---|---|---|---|
| La Azohía (Murcia, Spain) | 9.5±0.7 | 126±17 | 87.4 | 74.9 |
| El Campello (Valencia, Spain) | 11±4.0 | 133±27.1 | 98.9 | 67.2 |
| Cavalaire-sur-Mer (Provence, France) | 6.7±2.3 | 29±7.2 | 61.1 | 58.2 |
| Saleccia (Corsica, France) | 10±7.1 | 100±45.3 | 63.9 | 69.3 |
| Alassio (Liguria, Italy) | 7.5±0.7 | 10±4.3 | 40.9 | 33.2 |
| Marina di Pescia Romana (Lazio, Italy) | 8.5±2.1 | 40±12.7 | 63.1 | 47.9 |
| Mondello (Sicily, Italy) | 5±1.2 | 87±59.3 | 44.6 | 58.6 |
| Acharavi (Corfu Island, Greece) | 8.8±2.0 | 11±4.5 | 17.7 | 39.3 |

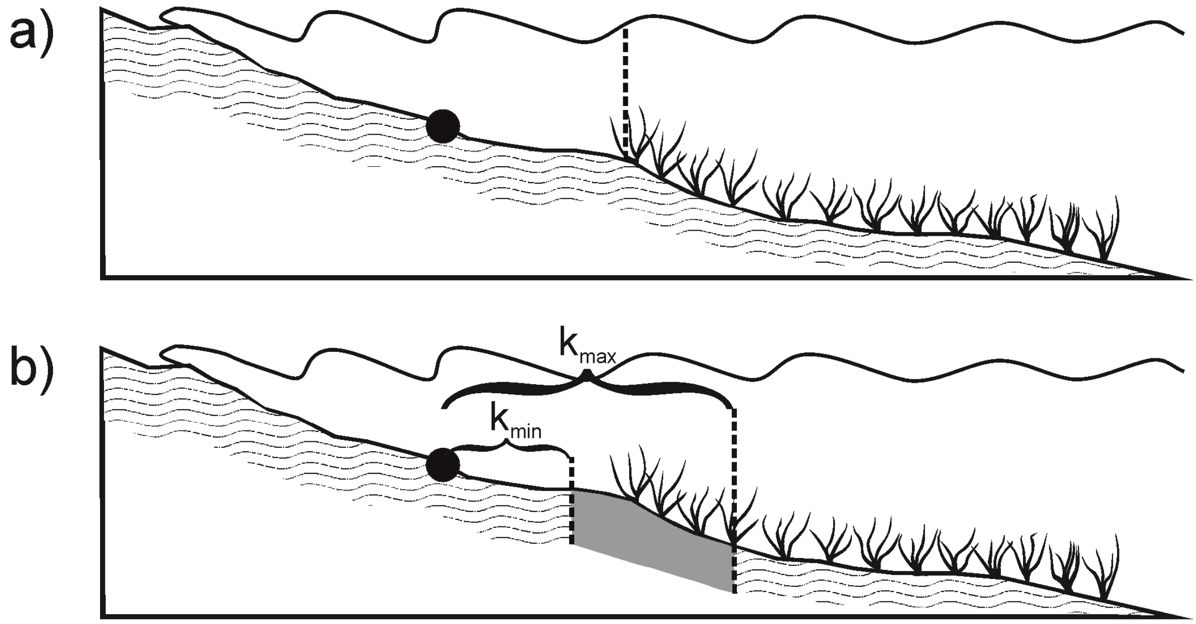
Table 4.

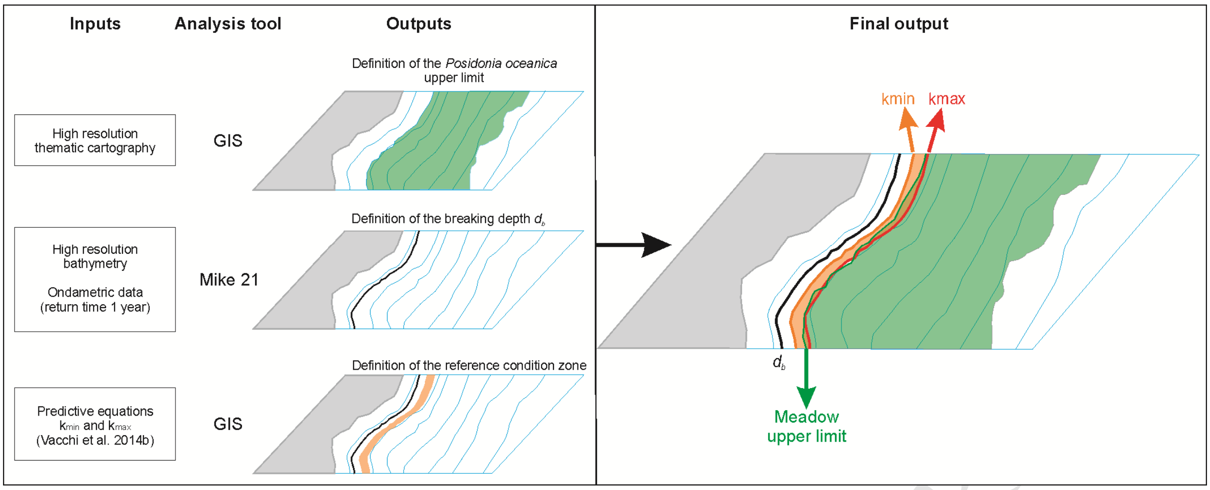
| Locality | Survey method | Map scale | Data sources |
|---|---|---|---|
| La Azohía (Murcia, Spain) | multibeam, aerial imageries | 1:10000 | 1, 2 |
| El Campello (Valencia, Spain) | multibeam, aerial imageries | 1:25000 | 1, 2 |
| Cavalaire-sur-Mer (Provence, France) | multibeam, aerial imageries | 1:10000 | 1, 3 |
| Saleccia (Corsica, France) | multibeam, aerial imageries | 1:10000 | 1, 4 |
| Alassio (Liguria, Italy) | side scan sonar, multibeam, aerial imageries | 1:10000 | 5 |
| Marina di Pescia Romana (Lazio, Italy) | multibeam, aerial imageries | 1:10000 | 6 |
| Mondello (Sicily, Italy) | multibeam, aerial imageries | 1:10000 | 7 |
| Acharavi (Corfu Island, Greece) | single beam, aerial imageries | 1:10000 | 8 |

Table 5.

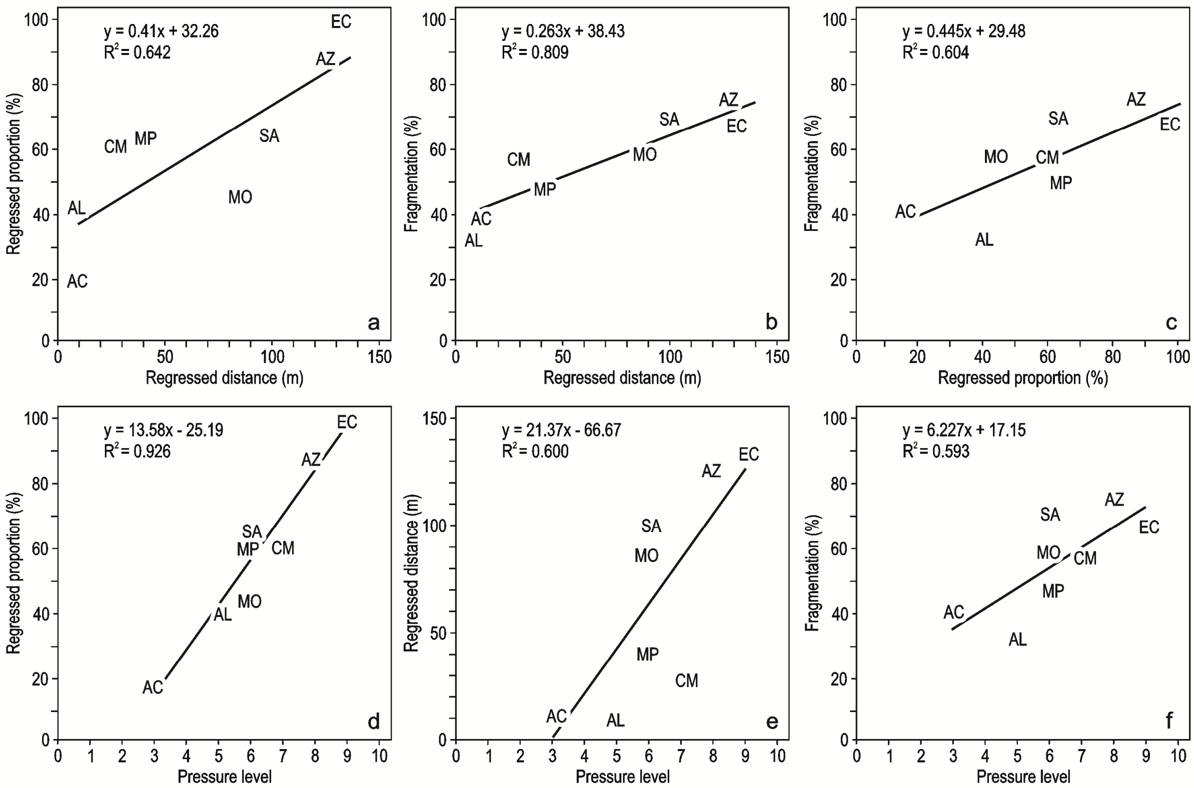
| **Locality** | **ε** | **$d_b$ (m)** | **$k_{min}$(m)** | **$k_{max}$(m)** |
|---|---|---|---|---|
| La Azohía (Murcia, Spain) | 632±542 | 5.7±0.5 | 190±158 | 277±222 |
| El Campello (Valencia, Spain) | 803±337 | 6.9±0.2 | 241±98 | 331±122 |
| Cavalaire-sur-Mer (Provence, France) | 152±41 | 4.3±0.1 | 50±11 | 79±19 |
| Saleccia (Corsica, France) | 262±228 | 7.0±0.6 | 82±66 | 125±93 |
| Alassio (Liguria, Italy) | 330±28 | 3.8±0.2 | 104±9 | 153±8 |
| Marina di Pescia Romana (Lazio, Italy) | 785±268 | 6.9±0.1 | 241±68 | 340±110 |
| Mondello (Sicily, Italy) | 237±32 | 3.2±0.2 | 75±9 | 115±13 |
| Acharavi (Corfu Island, Greece) | 1084±238 | 6.1±0.4 | 500±379 | 716±536 |


8°41' E
45°23' N
North
0
250 km
Alassio
Cavalaire-sur-Mer
Saleccia
Marina di Pescia Romana
El Campello
La Azohía
Mondello
Acharavi
Mediterranean Sea

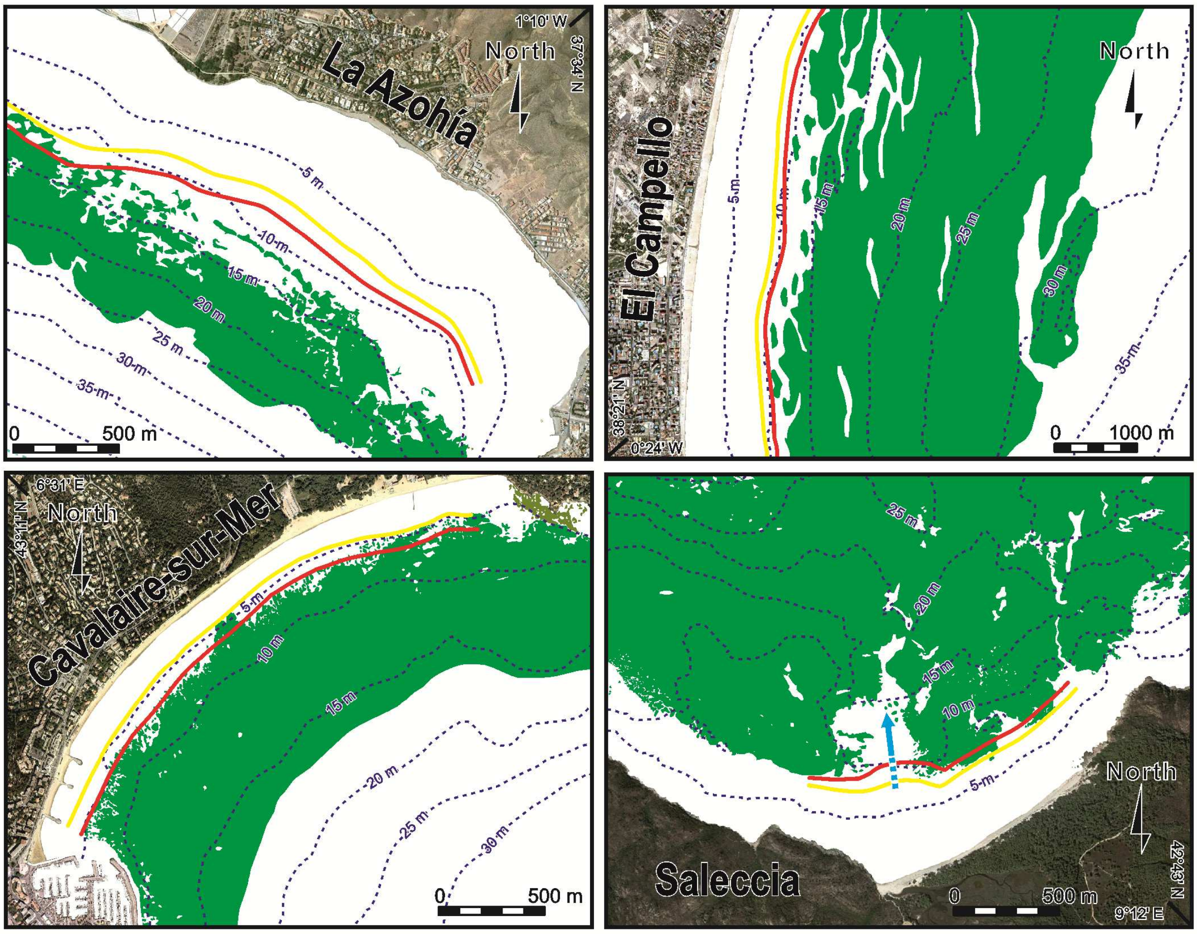

a)
b)
$k_{max}$
$k_{min}$

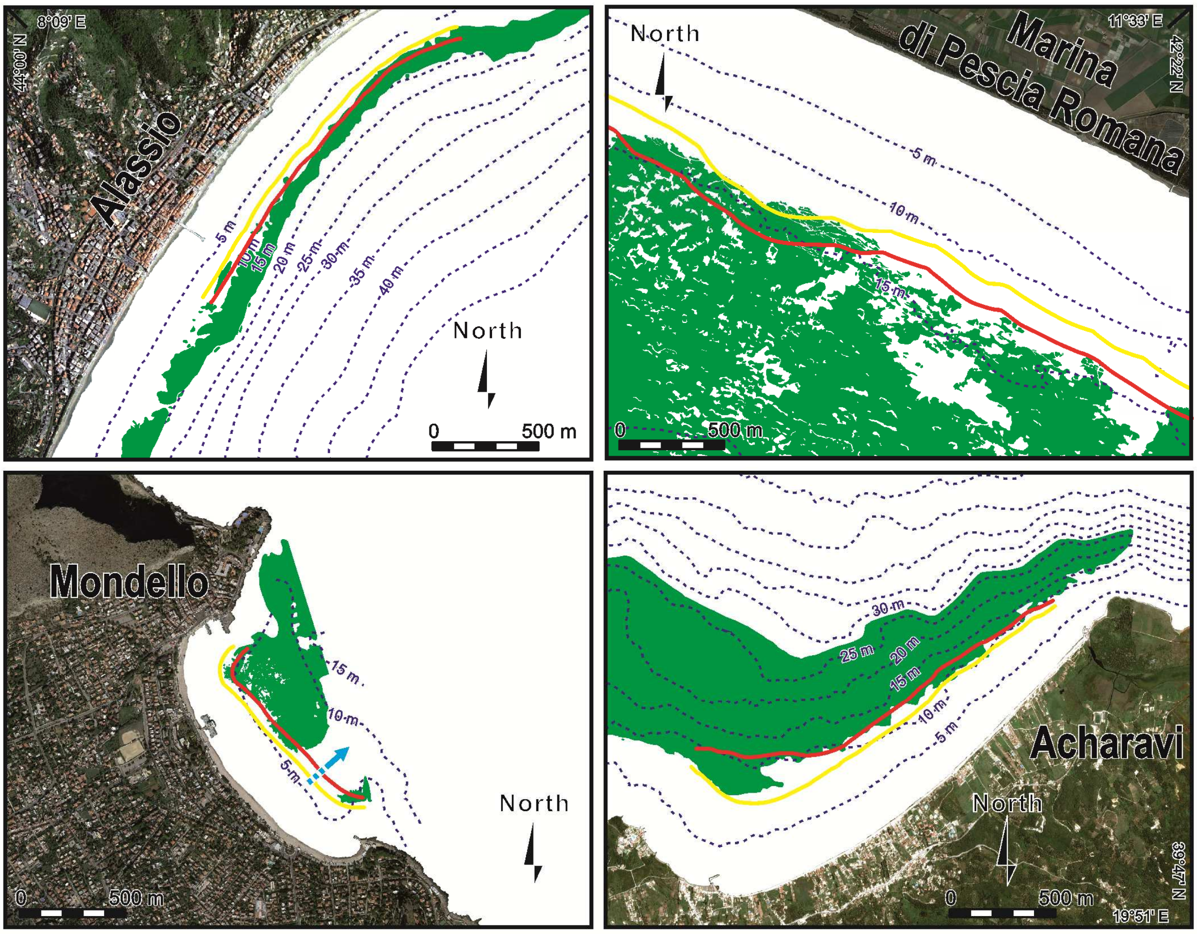


Inputs
Analysis tool
Outputs
Final output
Definition of the *Posidonia oceanica* upper limit
High resolution thematic cartography
GIS
Definition of the breaking depth $d_b$
High resolution bathymetry
Ondametric data (return time 1 year)
Mike 21
Definition of the reference condition zone
Predictive equations $k_{min}$ and $k_{max}$ (Vacchi et al. 2014b)
GIS
kmin
kmax
$d_b$
Meadow upper limit



y = 0.41x + 32.26
$R^2$ = 0.642
Regressed proportion (%)
Regressed distance (m)
EC
AZ
SA
CM
MP
MO
AL
AC
a
y = 0.263x + 38.43
$R^2$ = 0.809
Fragmentation (%)
Regressed distance (m)
AZ
SA
EC
CM
MO
MP
AC
AL
b
y = 0.445x + 29.48
$R^2$ = 0.604
Fragmentation (%)
Regressed proportion (%)
AZ
SA
EC
MO
CM
MP
AC
AL
c
y = 13.58x - 25.19
$R^2$ = 0.926
Regressed proportion (%)
Pressure level
EC
AZ
SA
MP
CM
MO
AL
AC
d
y = 21.37x - 66.67
$R^2$ = 0.600
Regressed distance (m)
Pressure level
EC
AZ
SA
MO
MP
CM
AC
AL
e
y = 6.227x + 17.15
$R^2$ = 0.593
Fragmentation (%)
Pressure level
AZ
SA
EC
MO
CM
MP
AC
AL
f


La Azohía
North
1°10' W
37°34' N
-5 m
10 m
15 m
20 m
25 m
30 m
35 m
0
500 m
El Campello
North
5 m
10 m
15 m
20 m
25 m
30 m
35 m
38°21' N
0°24' W
0
1000 m
Cavalaire-sur-Mer
North
6°31' E
43°11' N
5 m
10 m
15 m
20 m
25 m
30 m
0
500 m
Saleccia
North
25 m
20 m
15 m
10 m
5 m
42°43' N
9°12' E
0
500 m


Alassio
8°09' E
44°00' N
5 m
10 m
15 m
20 m
25 m
30 m
35 m
40 m
North
0
500 m
Marina
di Pescia Romana
11°33' E
42°22' N
North
5 m
10 m
15 m
0
500 m
Mondello
15 m
10 m
5 m
North
0
500 m
30 m
25 m
20 m
15 m
10 m
5 m
Acharavi
North
0
500 m
39°47' N
19°51' E

- Reference conditions are needed to measure long-term change in marine ecosystems
- We modelled the upper (landward) limit of seagrass meadows at Mediterranean scale
- All meadow upper limits appeared regressed and fragmented due to local disturbances
- Modelling represents an effective tool for evaluating seagrass meadow conditions